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## Whole-Farm Perspectives of Nutrient Flows in Grassland Agriculture

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### ABSTRACT

Grassland agriculture is an important industry for livestock production and land management throughout the world. We review the principles of nutrient cycling in grassland agriculture, discuss examples of grassland farming systems research, and demonstrate the usefulness of whole-farm simulation for integrating economic and environmental components. Comprehensive studies conducted at the Karkendamm experimental farm in northern Germany and the De Marke experimental farm in the Netherlands have quantified nutrient flows and developed innovative strategies to reduce nutrient losses in grassland farming systems. This research has focused on improving the utilization of manure nutrients on the farm by including grain crops in cropping systems with grassland and by incorporating manure handling techniques that reduce nitrogen losses. Although the information generated in experimental farms is not always directly applicable to other climates and soils, it is being transferred to other regions through computer simulation. A whole-farm model calibrated and verified with the experimental farm data is being used to evaluate and refine these strategies for commercial farms in other areas. Simulation of farms in northern Europe illustrate that on the sandy soils of this region, maize (*Zea mays* L.) silage can be used along with grasslands to increase farm profitability while maintaining or reducing nutrient loss to the environment. Use of cover crops, low emission barns, covered manure storages, and direct injection of manure into soil greatly reduces N losses from these farms, but their use creates a net cost to the producer. By integrating experimental farm data with whole-farm simulation, more sustainable grassland production systems can be cost-effectively evaluated, refined, and transferred to commercial production.

**G**RASSLAND AGRICULTURE is defined as “a farming system that emphasizes the importance of grasses and legumes in livestock and land management. Farmers

who plan row crops and livestock around their grassland hectares are grassland farmers” (Barnes, 1995). Before World War II, agriculture in the USA was very diverse and integrated, agricultural markets were primarily local, and nutrients were cycled mainly within farms and among local farms. With the advent of mechanization, chemical fertilizers, improved seeds, and agrichemicals, farm size increased, agricultural markets became national and international in scope, and nutrient cycles became more fragmented. Animal agriculture became specialized and concentrated, relying on off-farm sources of feeds and fertilizers, which resulted in nutrient accumulation on farms.

Similar changes in farm structure occurred in northwestern Europe during the 1960s and 1970s (de Wit et al., 1987). Farms located in regions with good soils were converted to crops like winter wheat (*Triticum aestivum* L.), oilseed rape (*Brassica napus* L.), potato (*Solanum tuberosum* L.), and sugarbeet (*Beta vulgaris* L.), whereas farms on poorer sandy soils specialized in milk production with permanent grassland as the main crop. These farms also intensified their use of purchased fertilizers and feed concentrates, and farm-scale nutrient budgets became less balanced because of low conversion rates of nutrients in milk and meat production (Van Keulen et al., 1996).

Nitrogen and phosphorus are the major farm nutrients of environmental concern. Nitrogen transforms to different compounds as it cycles through the farm and large losses may occur to the atmosphere, groundwater, and surface waters. The intensification of dairy farms in the Netherlands and northern Germany has created N surpluses of 150 to 250 kg N ha<sup>-1</sup> yr<sup>-1</sup>. This has resulted, among others, in increasing nitrate contamination of shallow groundwater with the highest nitrate concentrations in regions characterized by sandy soils and dominated by dairy farms (Spalding and Exner, 1993). Although there are fewer pathways for P loss, and losses are smaller relative to N, the environmental damage created by P-induced eutrophication of surface waters is an equally important concern.

As in any managed ecosystem, nutrient management in grassland agriculture must address multiple criteria, including air and water quality, nutrient use efficiency, and farm economics. Considering these multiple inter-

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**Abbreviations:** DAFOSYM, Dairy Forage System Model; DM, dry matter; IFSM, Integrated Farming System Model.

acting forces, simple component analysis is inadequate. A broader approach that addresses the whole farm is necessary. Although we recognize that nutrient effects are altered and manifested at larger scales such as watersheds, our frame of reference is on the scale at which nutrient management decisions are made—the farm scale.

The objectives of this paper are to: (i) briefly review the principles of nutrient cycling in grassland agriculture; (ii) present two examples of farm-level systems research dealing with nutrient cycling in grassland agriculture; and (iii) demonstrate how computer simulation with a whole-farm model provides a useful tool for integrating economic and environmental components.

### Principles of Nutrient Cycling in Grassland Systems

A nutrient cycle is the movement of an element among several reservoirs, or “pools,” in the environment. Nutrient cycles generally are not closed, except at a global scale; therefore, inputs and losses occur when viewed at the level of the microsite, field, farm, and region. Nutrients reside, more or less temporarily, in various reservoirs, including the plants and their residues, grazing or housed livestock, soil fauna and flora, and inorganic and organic compounds other than living tissue (Fig. 1). Pool sizes and composition vary with the nutrient considered. For example, inorganic forms of soil N are mainly exchangeable ammonium and solution nitrate, whereas the vast majority of inorganic P is adsorbed to, occluded within, or precipitated in compounds of calcium, aluminum, and iron, with very small concentrations in soil solution.

The addition of livestock to a farm increases the complexity and dynamics of nutrient cycling because of chemical and biological transformations that occur during digestion and after excretion (Jarvis et al., 1995). Furthermore, nutrient cycling in grazed grassland generally is more rapid and more heterogeneous than in me-

chanically harvested fields on mixed livestock or crop farms. Grazing livestock gather herbage from a large area, utilize a small portion of the nutrients, and excrete the remaining altered nutrient compounds in concentrated patches. For example, a dairy cow gathers 30 to 60% of the available forage from about 60 m<sup>2</sup> of pasture each day and then deposits most of the consumed nutrients in urine and dung patches covering only 2 to 3 m<sup>2</sup> (Haynes and Williams, 1993).

### Inputs

Nutrients enter a pasture as imported inorganic and organic fertilizers, stored manure, supplemental feed, symbiotic and nonsymbiotic N<sub>2</sub> fixation, and particulate, dissolved, and gaseous atmospheric deposition (Fig. 1). The presence and relative importance of each source depends on the nutrient in question, farm location, and farm management.

Atmospheric deposition of nutrients is generally considered to be small. Annual deposition of N in areas of the USA distant from local ammonia (NH<sub>3</sub>) sources are typically 1 to 6 kg N ha<sup>-1</sup>. In contrast, Burkart and James (1999) estimated annual depositions of 23 to 40 kg N ha<sup>-1</sup> over much of the U.S. Corn (maize) Belt, including low-level background loads and the redeposition of ammonia emitted from local sources (manure, inorganic fertilizer, and crop senescence). Fahey et al. (1999) estimated that annual bulk (wet and dry) deposition of ammonium near agricultural areas in New York state was 7 to 8 kg ha<sup>-1</sup>, twice the area's background level. Estimates of Burkart and James (1999) are consistent with reduced N deposition measurements in Western Europe (Ferm, 1998). A recent analysis indicated that estimated emissions in the USA are twice that of estimated deposition, which probably reflects inadequate sampling of deposition (Holland et al., 2005). Atmospheric sources of P generally are small (<1 kg P

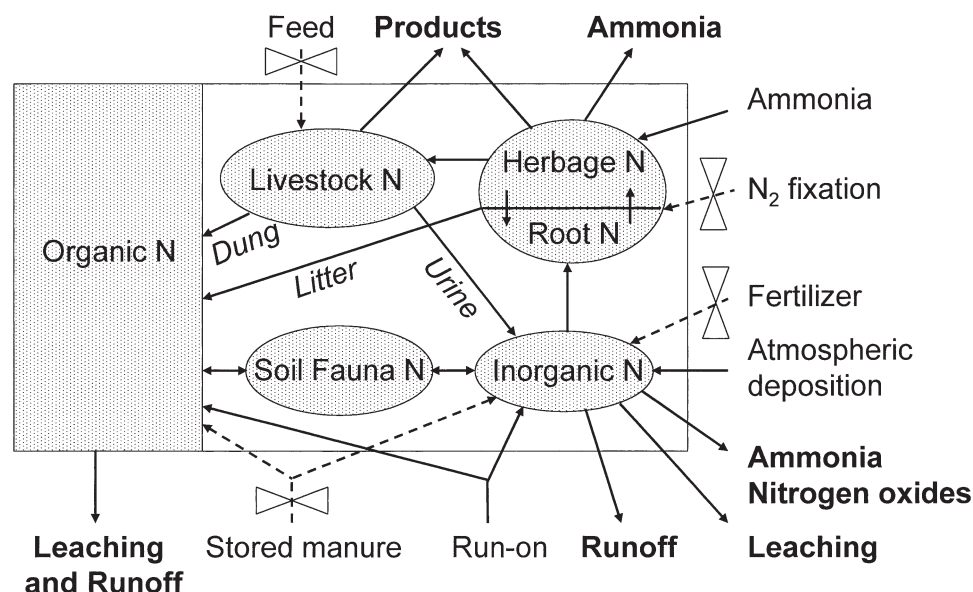


Fig. 1. Schematic N cycle showing inputs, outputs (bold-faced), and transformations in a pasture. Dashed arrows indicate inputs most easily managed. Some minor pathways are not included (e.g., organic N excretion in urine).

ha<sup>-1</sup>) in the absence of significant soil deposition by wind erosion (Goszczynska, 1983; Sharpley et al., 1985).

Nitrogen fertilizer recommendations on temperate grassland vary because of growing season length, water availability, and fertilization philosophy (Russelle, 1997; Vellinga et al., 2001). About one-half of the states in the USA surveyed in 1996 recommended lower N rates for pasture than mechanically harvested forage, while the others did not differentiate. As will be shown later in this paper, fertilizer rate often is directly related to nutrient loss. The availability of N is driven principally by biological activity and only secondarily by physics and chemistry (e.g., ammonia adsorption to clays, retention of nitrate against leaching in fine-textured soils), whereas P availability is driven primarily by chemical reactions, so P application rates can be based on soil test levels (Russelle, 1997).

Proper timing of fertilizer application can be important in promoting efficient utilization. Greater fertilizer N uptake by cool-season grasses occurs during spring growth than in fall growth (Stout and Jung, 1992). Luxury consumption during spring, however, can raise nitrate concentrations in forage with high fertilizer N application (Vetsch et al., 1999) and cause stand decline because of urine scorching (Lantinga et al., 1999). Tactical N fertilization, where N application rates and timing are adjusted according to measured soil nitrate at harvest, can reduce N surpluses (Titchen and Scholefield, 1992). This reduction, however, may come at the expense of animal and plant production (Laws et al., 2000), and it may not reliably reduce fertilizer N application requirements (Kowalenko and Bittman, 2000). To maximize economic returns from nutrient addition, management intensity must increase to utilize improved pasture production (Davison et al., 1985; Teitzel et al., 1991). Groot et al. (2003) concluded that long-term N losses can be reduced only by improving N use efficiency by both plants and livestock. Application of stored manure to perennial forages may increase yield and nutrient content, but excess soil P accumulation can be avoided by adjusting rates to achieve adequate P and fulfilling the remaining N requirement with inorganic N fertilizer (Evers, 2002).

Estimates of N<sub>2</sub> fixed by legumes in pasture range from 10 to 270 kg N ha<sup>-1</sup> (Ledgard, 2001), but most estimates fall between 100 and 200 kg N ha<sup>-1</sup> for typical forage legumes grown with grasses (West and Mallarino, 1996; Russelle, in press). As will be seen in the results below from northern Germany, N<sub>2</sub> fixation decreases as inorganic N supply to the legumes increases (Allos and Bartholemew, 1959). Some of the N fixed by legumes is transferred to companion nonlegumes, but the amount transferred is highly variable. Ledgard (1991) found that N transferred from white clover (*Trifolium repens* L.) to perennial ryegrass (*Lolium perenne* L.) was similar belowground (70 kg N ha<sup>-1</sup>) and through excreta (60 kg N ha<sup>-1</sup>). Together these transfers represented nearly one-half of the total N<sub>2</sub> fixed by the clover (270 kg N ha<sup>-1</sup>).

The proportion of legume in a sward necessary to provide sufficient N to the nonlegume varies with le-

gume species and forage utilization by the livestock, which depends on stocking rate, grazing management, and forage palatability. In New Zealand, pasture yields were similar when white clover comprised 9 to 30% of the sward (Ledgard, 1991). At low to moderate N rates, pasture yield can increase without reduction in legume stands (Whitehead, 1995), but this response is affected by soil moisture conditions, grazing intensity, and other factors that affect competition for limiting resources. Legume populations fluctuate over time and across the landscape, predominating in mixtures with non-N<sub>2</sub>-fixers where soil N supply is low and P supply is adequate (Steele and Shannon, 1982; Schwinning and Parsons, 1996; Loiseau et al., 2001). In the Netherlands, mixed swards of grass and white clover produced 85% of the milk yield per hectare compared with grass fertilized with N (Schils, 2002). Farm-level N utilization was about 25% in both systems with no difference in the average nitrate concentrations in drainage water. Total energy use in the clover-based system was 15% lower than that of the inorganic fertilizer-N system.

Supplemental feeding increases nutrient import to a pasture. For a dairy herd with a feed-to-milk N use efficiency of 22% that spends 20 h d<sup>-1</sup> in pasture, 65% of the N from the supplemental feed is deposited in the pasture (van Vuuren and Meijs, 1987). The net effect is moderated where supplementation increases animal production and reduces both total and labile N and P excretion (Valk and Hobbelink, 1992; Ebeling et al., 2002).

## Transformations and Loss

The only desirable nutrient "loss" from the farm is via products, such as milk, meat, wool, and feeds sold. Retention of consumed feed nutrients in ruminant animal products is low, ranging between 5 and 30%, with lower retention rates in animal tissue and higher rates in milk. The importance of other loss pathways depends on the specific situation. For example, nitrate N leaching and N and P runoff can impair water quality, whereas gaseous losses of N adversely affect chemical and physical properties of the atmosphere. The amount of nutrient loss often increases with production intensity, because of inherent limits to the efficiency of milk and meat production (Watson et al., 1992). Furthermore, when N loss by a given pathway is reduced, losses by one or more remaining pathways may increase because N transformations and losses are concentration-dependent (Brink et al., 2001). Once production is maximized in systems near equilibrium with respect to soil organic N, all further N inputs are lost to the environment (i.e., output = input; Fig. 2). This is consistent with research in production systems without livestock (Kolenbrander, 1981). As is discussed later, however, net N loss at a given N input rate will decrease directly if more N efficient production of animal or crop products is achieved. The situation with P loss is more complicated, but runoff losses typically increase with input rate or soil test level (Sharpley et al., 2002) and if runoff occurs soon after fertilization or grazing (Nash et al., 2000). Leaching

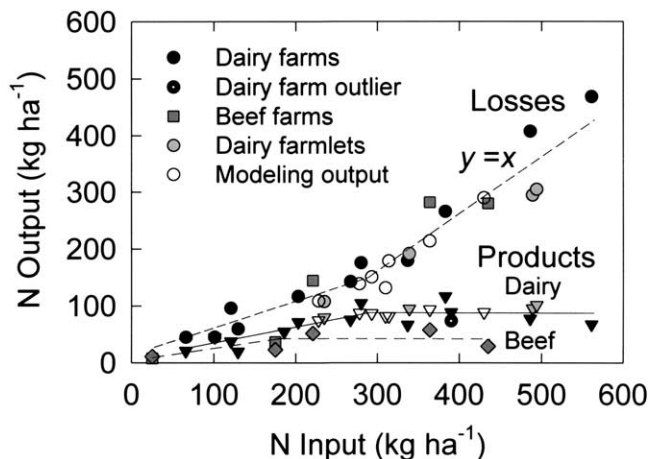


Fig. 2. Relationship between total N input and N output as products (milk, meat, and crops) and as losses (volatilization, denitrification, runoff, leaching, and transfer to unproductive areas) in dairy and beef production systems that involve some grazing. Data from on-farm and replicated large plot research in New Zealand, Europe, and the USA (Kaffka and Koepf, 1989; Granstedt, 1995; van der Werf et al., 1995; Klausner et al., 1998; Goh and Williams, 1999; Ledgard et al., 1999; Watson and Atkinson, 1999; Hilhorst et al., 2001; Table 4 in this paper) and modeled output for farms in northern Germany (Table 5 this paper) and De Marke (Table 6 in this paper). One data point for dairy (closed, dotted circle) likely underreported leaching losses.

losses can be significant where organic amendments are applied (Jensen et al., 1999) and where P sorption capacity is low (Burkitt et al., 2004).

The major N reservoir in the soil is organic matter. This N pool is typically larger and mineralizes faster under productive grassland than under annually cropped land (Whitehead, 1995). Broadly speaking, annual net N mineralization of soil organic matter N is 5 to 9% in temperate pastures (Hatch et al., 1991) compared with 2 to 3% for annual crop systems (Schepers and Mosier, 1991). Livestock accelerate nutrient cycling directly through decomposition and excretion of plant-derived nutrients and indirectly through the effects of grazing and excreta on soil biota (Bardgett and Wardle, 2003). Microbial decomposers account for up to 70% of the net primary production in temperate pastures (Hutchinson and King, 1982). Much of the plant residue remaining after grazing and a large amount of root biomass decompose and release nutrients. In a grazed temperate grassland producing 10 Mg herbage DM ha<sup>-1</sup>, about 180 kg N ha<sup>-1</sup> was returned to the soil in nonutilized herbage, with an additional 105 kg N ha<sup>-1</sup> returned in decaying roots (Whitehead, 1986). Root turnover is promoted by environmental stresses such as shading (Butler et al., 1959), by hoof damage (Marriott and Smith, 1992), predation by insects (Murray and Clements, 1998), and plant diseases (Bardgett et al., 1999).

Dung contains undigested herbage residues, products of animal metabolism, ingested soil, and a large biomass of microorganisms (Kirchmann and Witter, 1992). Nearly all excreted P is in dung, while N excretion in dung is relatively constant at about 8 g kg<sup>-1</sup> of feed consumed. Remaining excreted N is in urine, so the portion of N excreted in urine increases with dietary

protein concentration. Organic P in dung is relatively constant at 0.6 g kg<sup>-1</sup> feed consumed, with the remainder being mainly dicalcium phosphate (Barrow, 1987; Haynes and Williams, 1993).

Nitrogen losses from urine spots are higher than those from dung. Nitrogen in dung is only 20 to 25% water-soluble, with little free NH<sub>3</sub> (Kirchmann and Witter, 1992), so volatile loss of NH<sub>3</sub> is generally less than 5% (Ryden et al., 1987). In contrast, urinary N is 50 to 80% urea, which is hydrolyzed rapidly by the ubiquitous urease enzyme (Haynes and Williams, 1993). Urine usually has a pH above 7.4 for animals fed forage diets (Whitehead et al., 1989), and urea hydrolysis raises soil pH by 1 to 3 units in the uppermost layers before declining as ammonium (NH<sub>4</sub><sup>+</sup>) is nitrified (Vallis et al., 1985). Nitrification is delayed by high pH, NH<sub>4</sub><sup>+</sup>/NH<sub>3</sub> concentration, and osmotic strength in the urine spot (Monaghan and Barraclough, 1992). As a result, NH<sub>3</sub> volatilization losses of 10 to 25% of applied urinary N are common on medium-textured soils in temperate regions (Lockyer and Whitehead, 1990) and can reach 90% in semiarid regions (Woodmansee, 1978). Ammonia losses are generally lower on sandy soils because of greater infiltration and on soils with high clay and organic matter contents because of NH<sub>4</sub><sup>+</sup> adsorption on cation exchange sites (Whitehead and Raistrick, 1993).

Ammonia loss from urine spots increases under high N fertilization because more N is excreted in urine (Jarvis, 1990), and remaining plant shoots have less capacity to absorb the volatilized NH<sub>3</sub> (Lemon and van Houtte, 1980). Ammonia volatilized from the soil surface can be recycled rapidly when absorbed by herbage because NH<sub>3</sub> exchange between the canopy and the atmosphere is regulated by the canopy NH<sub>3</sub> compensation point (Denmead et al., 1976; Langford and Fehsenfeld, 1992). Taller swards also absorb more NH<sub>3</sub> because of a greater residence time for the volatilized NH<sub>3</sub> within the canopy (Sommer and Hutchings, 2001). When wheat leaf area index was about 5, NH<sub>3</sub> absorption was 0.74 g N m<sup>-2</sup> leaf surface, resulting in a 60% reduction in NH<sub>3</sub> volatile loss from applied manure slurry (Sommer et al., 1997). In contrast, these plants emitted NH<sub>3</sub> during stem elongation and absorbed none of the volatilized manure NH<sub>3</sub>. Volatile NH<sub>3</sub> loss from pastured livestock usually is less than the combined loss from the barn, manure storage, and field application in confinement dairy systems (Whitehead, 1995).

Chemical conditions in urine spots may cause temporarily high nitrite concentrations (Burns et al., 1995). Lower activity of nitrite oxidizers under these conditions (Smith et al., 1997) leads to losses of N oxides [primarily nitric and nitrous oxide (N<sub>2</sub>O)] formed during nitrification (Clough et al., 2003). The large increase in soil pH as urea is converted to NH<sub>4</sub><sup>+</sup> increases the concentrations of dissolved organic carbon and total dissolved P in soil solution (Shand et al., 2000). Higher dissolved organic carbon availability and possible nitrate formation are two conditions that support denitrification in urine spots (Chantigny, 2003). In one study, urine spots on 10 to 15% of the pasture area contributed more to the total denitrification than the remaining area in a grazed pe-

rennial ryegrass–white clover pasture (Ruz-Jerez et al., 1994).

At low to moderate N fertilization rates, soil nitrate concentrations in grass-based pastures may be too low to support large losses of N by denitrification (Parsons et al., 1993; Parsons and Keller, 1995). Thus, addition of nitrate can greatly stimulate denitrification (Groffman et al., 1993; Parsons et al., 1993). This suggests that inorganic fertilizer N additions to pastures on wet soils should be made with ammonium-based materials and in small doses. Annual loss through  $\text{NH}_3$  volatilization and denitrification from a temperate pasture grazed by dairy cows in south-eastern Australia was  $23 \text{ kg N ha}^{-1}$  from paddocks without N fertilizer and two and three times higher with ammonium nitrate and urea fertilizer, respectively, applied at  $200 \text{ kg N ha}^{-1}$  (Eckard et al., 2003). A few instances of wet soil conditions soon after N fertilization or grazing resulted in about one-half of the total  $\text{N}_2\text{O}$  emission from an Irish pasture (Scanlon and Kiely, 2003).

Soil nitrate is also subject to leaching loss. Leaching losses are smaller under perennial forages harvested by mowing than under annual crops because cool-season perennials have an extended growing season, high dry matter yield (high N uptake), and high water use. In Wisconsin, alfalfa (*Medicago sativa* L.)–bromegrass (*Bromus inermis* Leyss.) fields used 55 to 65 cm of water per year, whereas maize and small grains used only 45 to 55 cm and bluegrass (*Poa pratensis* L.) turf used 30 to 45 cm (Schulte and Walsh, 1993). As a result of greater water use and nitrate consumption, alfalfa lost less than  $11 \text{ kg N ha}^{-1}$  through leaching to tile drains over a 6-yr period in southwestern Minnesota, whereas maize–soybean [*Glycine max* (L.) Merr.] rotations lost nearly  $500 \text{ kg N ha}^{-1}$  (Randall et al., 1997). Higher nitrate losses occurred under grazing than mowing at the same applied fertilizer N because excreta patches provided more N than plants could absorb within and around the patch area (Garwood and Ryden, 1986).

Nitrate leaching from grasslands increases with N input, whether that input is from symbiotic  $\text{N}_2$  fixation or N fertilizer (Ledgard, 2001). Sward N uptake is a key determinant in reducing soil solution nitrate concentration. Therefore, nitrate losses often increase after dry periods (Scholefield et al., 1993; Stout et al., 2000) and in winter (Owens et al., 1994). The likelihood of leaching depends on climate, soil, and plant characteristics, with the highest losses in humid climates (Kellogg, 2000) or irrigated systems, on coarse-textured soils or soils with artificial drainage, and under plants with short root systems, such as perennial ryegrass and white clover (each with active root zone depths restricted to about 50 cm). Little nitrate leaching occurred on fine-textured soils under pastures in the Upper Midwest USA with low to moderate inorganic N fertilizer application rates (about  $50 \text{ kg N ha}^{-1}$  annually) or in legume-grass mixtures (Russelle et al., 1997; Russelle, 2003). Although fall-applied dairy manure slurry and ammonium fertilizer improved pasture production to the same extent, nitrate leaching losses were much higher for inorganic fertilizer (Di et al., 1999).

In addition to nitrates, dissolved organic compounds may comprise a large fraction of the total N leaching loss in grasslands (McCarthy et al., 1996; Bhogal et al., 2000), as observed on sandy soils in northern Germany (next section). Leaching of P to tile drains also may be significant in manured or pastured conditions (Sims et al., 1998; Hooda et al., 1999).

Application of stored manure to grassland may increase nutrient losses (Gillingham, 1987; Cabrera and Gordillo, 1995). Ammonia volatilization can be reduced by surface banding (Bittman et al., 2002) or slurry injection, as used on the Dutch De Marke farm discussed later, and runoff and leaching can be minimized by avoiding areas of high nutrient status. Reducing the surface area of manure that is exposed to the atmosphere greatly reduces ammonia volatilization (Sommer and Hutchings, 2001), but injection raises  $\text{CO}_2$  emissions because more tractor fuel is needed to compensate for the increased draft (Hansen et al., 2003). Increases in  $\text{N}_2\text{O}$  emissions and nitrate leaching can be expected with reductions in  $\text{NH}_3$  volatilization, if N use efficiency is not increased concurrently (Brink et al., 2001).

Pastures are often the preferred crop on hilly landscapes. Pastures help reduce N and P loss in sediment and runoff water compared with annual crops because of faster water infiltration rates, less overland flow, and more stable soil aggregates (Zemenchik et al., 2002). Exceptions occur where livestock have damaged the plant cover or soil structure (Pearson and Ison, 1987), nutrient enrichment near the soil surface is excessive (Correll, 1996), or nutrients are easily leached from herbage (e.g., frozen or dry vegetation; Schuman et al., 1973).

### Spatial Heterogeneity

Sheep and steers excrete 85 to 95%, and lactating dairy cows excrete 70 to 80%, of the N they consume (Haynes and Williams, 1993). Although excreta patches cover only 14 to 22% of a pasture with a stocking rate of one dairy cow per hectare, at least twice this area is affected by the excreta because of plant uptake of excreta nutrients, altered plant competition, altered animal feeding preference, and redistribution of dung by fauna (Mathews et al., 1996).

In general, dairy cows apply 500 to  $1200 \text{ kg N ha}^{-1}$  in urine spots with a liquid application rate equivalent to  $250 \text{ cm h}^{-1}$  of rainfall (Steele, 1987; Haynes and Williams, 1993). Macropore flow facilitates deep infiltration of some urine into the soil, but most N remains in the upper 20 cm. Dung patches contain the equivalent of up to  $1000 \text{ kg N ha}^{-1}$  and  $280 \text{ kg P ha}^{-1}$ .

Grazing animals do not distribute excreta randomly across a pasture. A higher amount of excretion occurs where animals spend more time, such as in shade, near gateways and drinking water, and in areas sheltered from the wind. In New Zealand hill pastures, greater dung deposition on hill crests resulted in P return that exceeded forage uptake, whereas P return on steep slopes was only 30% of the forage uptake (Rowarth et al., 1992).

This larger-scale patchiness of nutrient buildup in pastures affects both fertilizer requirements to optimize productivity and the location of critical source areas with disproportionate nutrient loss (Gburek and Sharpley, 1998). Areas of the pasture near drinking water sources or shade may have 2 to 15 times the topsoil P and K levels of other areas (Schomberg et al., 2000; West et al., 1989). Animal behavior can be altered in relatively intensive management systems through the placement of water, use of small paddocks, etc. (Mathews et al., 1996). In instances where animal behavior cannot be managed to achieve more even excreta distribution, such as in extensive grazing systems or those on rugged landscapes (Gillingham, 1987), strategic placement of shade, water, or supplemental feed or minerals may be used to facilitate nutrient buildup on land where the risk of runoff to surface water is smallest.

Even with good animal management, tremendous variation in soil fertility exists in pastures. Differences in soil nitrate concentration of  $90 \text{ mg kg}^{-1}$  were found in samples taken 4 cm apart (Thompson and Coup, 1940). Spatially-discrete soil sampling schemes may help avoid over-application of fertilizer nutrients. Especially in intensive, high fertility pastoral systems, additional nutrients should not be applied to areas where excreta are concentrated. However, simply withholding fertilizer N from urine spots and applying  $180 \text{ kg N ha}^{-1}$  to the remaining area did not reduce average soil nitrate concentrations (Cuttle et al., 2001), indicating that N rates may need to vary in smaller increments. A more practical approach than intensive soil sampling may be to use spectral reflectance of herbage in combination with real-time variable-rate N fertilization, which maintained yields with 40% less N fertilizer in a single-species pasture (Taylor et al., 1998).

### Farming Systems Research

Nutrient management at the farm scale is complex. A suitable balance must be made between minimizing nutrient losses and maintaining production costs to provide sustainable production systems. Limited information exists on the management and monitoring of nutrient flows in grassland agriculture at the whole-farm or systems scale. Two recent experimental research efforts have focused on nutrient management in whole-farm systems where grassland is a major component.

### Managing Grassland Farms for Reduced Nitrogen Loss in Northern Germany

Multiple interactions influencing nitrogen fluxes in the soil–plant–animal system were studied at the Karkendamm experimental farm (part of the University of Kiel) near Hamburg, Germany (Taube and Wachendorf, 2001). The goal was to enhance N use efficiency in dairying on the well-drained sandy soils (pH of 5.0–5.5) commonly found in this region.

Experiments were established on permanent grassland (1.8 ha in total) to compare grassland management systems for white clover performance, N losses via leaching and denitrification, along with yield and quality of

herbage (Trott et al., 2004; Wachendorf et al., 2004a; Lampe et al., 2004). The systems included cutting (i.e., mechanical harvest for hay or silage), grazing, and two mixed systems of cutting and grazing all with various levels of mineral fertilizer (0, 100, 200, 300, and  $400 \text{ kg N ha}^{-1}$ ), and slurry application (0 or  $20 \text{ m}^3 \text{ ha}^{-1}$ ). Slurry diluted with water ( $2.4 \text{ kg N m}^{-3}$ ) was applied once in spring via a drag hose technique to reduce ammonia loss. These grassland management systems predominate in northern Germany.

A novel aspect of the Karkendamm research was the calculation of  $\text{CO}_2$  environmental loads per unit of energy (net energy of lactation,  $\text{NE}_L$ ) in the feed produced. The  $\text{CO}_2$  loads were an additional environmental indicator to demonstrate that focusing only on nutrient use efficiency in grassland management might inadvertently cause other environmental problems.

### White Clover Contribution, Sward Productivity, and Net Energy Yields

Across all grassland management systems, an increase in total N supply (including N from mineral fertilizer, slurry, atmospheric deposition, excreta, and rejected herbage) reduced white clover proportion in grassland (Fig. 3; Trott et al., 2004). Less clover grew in grazed than in cut swards, primarily because of the greater amount of N returned in animal excreta along with selective grazing and perhaps treading damage. Although clover proportions in swards cut once in spring and then grazed were similar to those that were grazed only, more clover was found in swards cut twice (spring and early summer) followed by grazing. Nitrogen fixation by white clover was directly related to biomass yield, so more occurred in the cutting systems than under grazing. The mean annual  $\text{N}_2$ -fixation rates ranged from 0 to  $166 \text{ kg N ha}^{-1}$ .

Overall mean annual net energy yields (expressed in terms of  $\text{NE}_L$ ) were  $48 \text{ GJ NE}_L \text{ ha}^{-1}$  in systems using grazing only,  $54 \text{ GJ NE}_L \text{ ha}^{-1}$  for one cutting followed

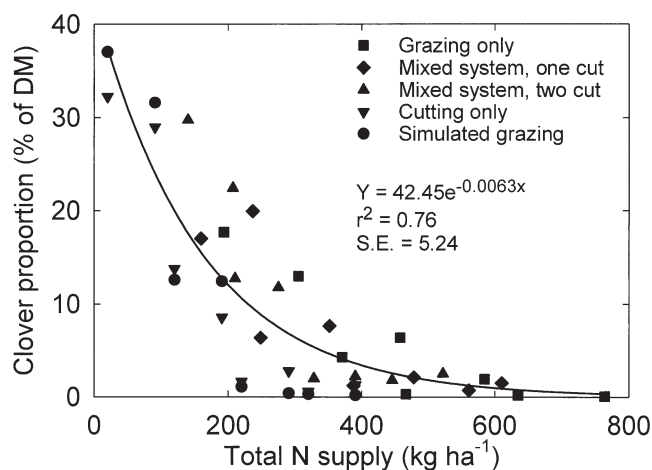


Fig. 3. The effect of total N supply (mineral fertilizer N + slurry N + N deposition + excreta N + residue N) on the proportion of white clover under different defoliation systems (mixed systems are one cut or two cuts in spring with subsequent grazing). Average of 1997 to 2001 at the Karkendamm experimental farm in northern Germany (Trott et al., 2004).

by grazing (mixed system I), 57 GJ NE<sub>L</sub> ha<sup>-1</sup> for two cuttings followed by grazing (mixed system II), 53 GJ NE<sub>L</sub> ha<sup>-1</sup> for cutting only, and 62 GJ NE<sub>L</sub> ha<sup>-1</sup> for simulated grazing (clipped to represent grazing without animals), respectively. Yields increased linearly with increasing total fertilizer N applied. Slurry application improved production under grazing only and Mixed System I, with an average response of 8.3 and 4.6 GJ NE<sub>L</sub> per 20 m<sup>3</sup> of applied slurry, respectively, whereas under simulated grazing, cutting only, and Mixed System II no slurry effect was measured. The production of grazed swards was significantly lower than that of the corresponding simulated grazing treatments, with an average annual difference of 4.9 GJ NE<sub>L</sub> (0.8 Mg DM) ha<sup>-1</sup>. This was likely due to the low utilization efficiency of excreta N, reduced clover proportions, and urine scorching which often occurs on sandy soils.

### Nitrogen Surplus

The N surplus across all treatments was linearly related to total N supplied (Fig. 4a; Trott et al., 2004).

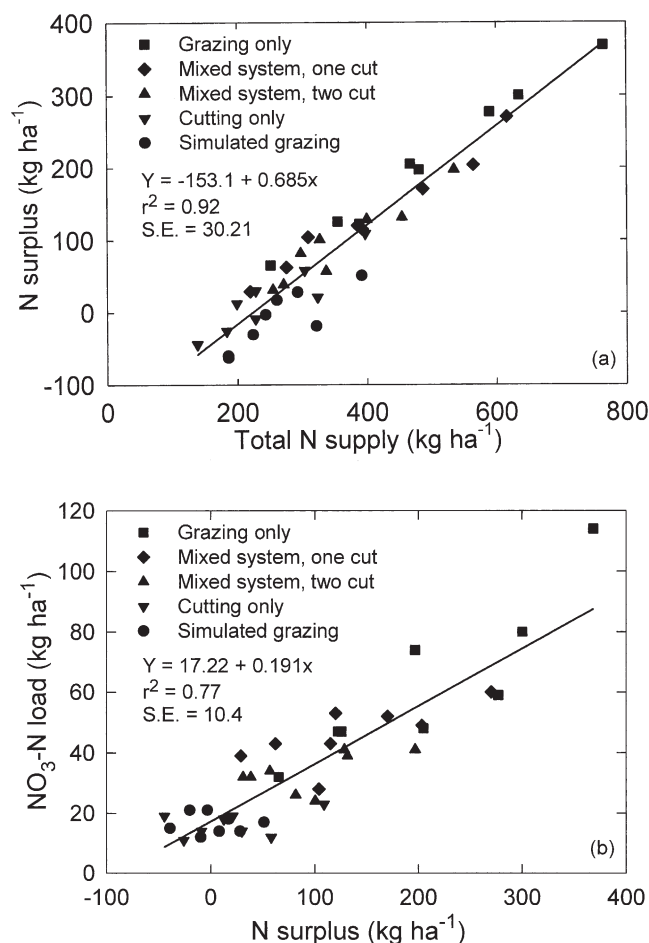


Fig. 4. (a) The effect of total N supply (mineral fertilizer N + slurry N + N deposition + excreta N + residue N) on annual N surplus under different defoliation systems (mixed systems are one cut or two cuts in spring with subsequent grazing) at the Karkendamm experimental farm in northern Germany (Trott et al., 2004). (b) The relationship between annual N leaching losses and annual N surplus on grassland at the Karkendamm experimental farm in northern Germany (Wachendorf et al. (2004a).)

Negative N balances occurred for the cutting and simulated grazing treatments because N from the mineralization of soil organic N was not included in the calculated balance. This N obtained from the soil was estimated from unfertilized clover-free plots to average 69 kg ha<sup>-1</sup> yr<sup>-1</sup> under the cutting only treatment. Surpluses were relatively high with applied slurry because of low recovery of slurry N in herbage and high NH<sub>3</sub> volatilization after surface spreading. In the grazing only treatment, N inputs exceeded N outputs at all levels of N fertilization. The increase in N surplus per kilogram N applied was 2.5 times higher under grazing than cutting. Thus, to reduce N surpluses in rotational stocking systems, less N must be applied. The inclusion of a silage harvest with rotational stocking systems reduced N surplus because more N was removed from the field in the conserved forage. Mixed System II provided high yields with a moderate N application rate. Cutting-only systems allowed N application rates beyond 300 kg ha<sup>-1</sup> with an increased energy yield and more cost-effective use of N.

### Nitrogen Losses

Nitrate leaching losses (determined with ceramic suction cups) were strongly affected by the type of defoliation, with the lowest values in the cutting and simulated grazing systems (Wachendorf et al., 2004a). In these systems, nitrate concentrations in the leachate generally were below the European Union threshold of 50 mg NO<sub>3</sub> L<sup>-1</sup> (11.3 mg N L<sup>-1</sup>), which is equivalent to 23 kg N ha<sup>-1</sup> leached at an average drainage of 205 mm of soil solution. The highest concentrations of up to 250 mg NO<sub>3</sub> L<sup>-1</sup> (56 mg N L<sup>-1</sup>) were measured in the grazed-only treatment, which corresponded to a leaching loss of up to 114 kg N ha<sup>-1</sup> yr<sup>-1</sup>. Intermediate nitrate losses occurred in the mixed systems. In all systems, nitrate losses increased with increasing N input. In defoliation systems using grazing, leachate nitrate contents were well above the European Union drinking water limit.

Leaching loss and N surplus were positively related (Fig. 4b). Leaching losses occurred under all systems, even where N surpluses were negative. Other than the soil organic N source that was not included in the balance, negative N surpluses may be due to underestimation of biological N-fixation (Neuendorff, 1996; Loges, 1998). The regression in Fig. 4b implies that N surpluses of not more than 30 kg ha<sup>-1</sup> are acceptable to meet the European Union standard for drinking water.

There is some evidence that leaching of dissolved organic N contributes significantly to overall N losses on sandy soils (McCarthy et al., 1996). In the Karkendamm experiment, dissolved organic N accounted for 50% of the total N leached in the cutting only treatment during autumn and winter (Christine Wachendorf, Univ. of Hamburg, 2004, personal communication).

Total N<sub>2</sub>O emissions measured from the soil surface over an 11-mo period (measured with ventilated chambers) ranged from 1.7 to 4.9 kg N ha<sup>-1</sup>. The lowest N<sub>2</sub>O emissions occurred with an application of 100 kg mineral N ha<sup>-1</sup> and the highest emissions occurred with both

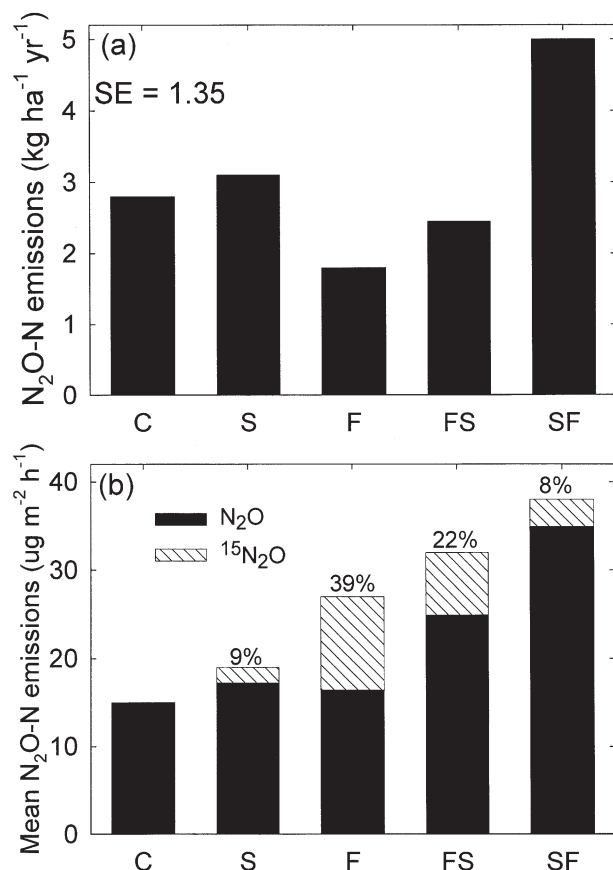


Fig. 5. (a) Total  $\text{N}_2\text{O-N}$  emissions from April 2001 to March 2002 and (b) mean  $\text{N}_2\text{O-N}$  emission rates for the cutting period (April to July) and fertilizer-borne  $\text{N}_2\text{O}$  (%) at the Karkendamm experimental farm in northern Germany (Kelm et al., 2004). C = Control; S =  $^{15}\text{N}$ -labeled slurry; F = 100 kg N  $\text{ha}^{-1}$  labeled with  $^{15}\text{N}$ ; FS = nonlabeled slurry + 100 kg N  $\text{ha}^{-1}$  labeled with  $^{15}\text{N}$ ; SF =  $^{15}\text{N}$ -labeled slurry + nonlabeled 100 kg N  $\text{ha}^{-1}$ .

slurry (74 kg N  $\text{ha}^{-1}$ ) and mineral fertilizer (100 kg N  $\text{ha}^{-1}$ ) applied (Lampe et al., 2004). Differences among treatments were not significant (Fig. 5a), which may have been due to the heterogeneous distribution of excrement and the  $\text{N}_2$  fixation of white clover providing a similar N status in all treatments. However, during April to July mean  $\text{N}_2\text{O}$  emission rates increased significantly with increasing N input (Fig. 5b). Differences among the fertilized treatments and the unfertilized control indicate that a large part of the mineral N fertilizer was rapidly available for microbiological processes, whereas with slurry application only the ammonium N was available (about 50% of the total slurry-N). Emissions in winter, primarily during freezing and thawing cycles, made a considerable contribution to the total annual  $\text{N}_2\text{O}$  loss. Flux rates varied between 17 and 147  $\mu\text{g N}_2\text{O-N m}^{-2} \text{ h}^{-1}$ , suggesting that thawing enhanced microbial activity because of increased carbon availability from microorganisms killed by freezing under wet soil conditions (Christensen and Christensen, 1991).

#### Energy Efficiency and Carbon Dioxide Emissions

Nitrogen use efficiency and loss must be considered along with other environmental concerns such as energy

use and  $\text{CO}_2$  emissions. The efficiency of fossil energy use was determined as the feed net energy yield per unit of fossil energy input in production activities. Energy inputs included both direct (diesel use for field operations) and indirect (fossil energy input in the manufacture and distribution of fertilizers, pesticides, machinery, seeds, etc.) inputs (Kelm et al., 2004).

Net energy yields for white clover and grass swards showed a relatively weak and linear response to increasing N fertilizer application; thus, energy efficiency declined with increasing mineral N fertilizer input (Kelm et al., 2004). This effect was most pronounced on pastures because mineral N fertilizer constituted a larger proportion of total energy input compared with cutting-only and mixed cutting and grazing systems. Except for very low yields of less than 50 GJ  $\text{NE}_L \text{ ha}^{-1}$ , where pasture was the most energy-efficient system, a given net energy yield was produced most energy-efficiently in Mixed System I where additional yield compensated for the higher energy input from increased machinery activities.

Carbon dioxide emissions showed similar trends as found for energy efficiency (Kelm et al., 2004). The  $\text{CO}_2$  emission factors per unit of energy use for the largest energy inputs in forage production (diesel fuel and mineral N fertilizer) were similar (82.6 and 81.0 kg  $\text{CO}_2 \text{ GJ}^{-1}$ , respectively), so  $\text{CO}_2$  emissions were nearly proportional to energy use.

The benefit of reduced nitrate leaching loss from cutting systems must be considered along with the significantly lower energy efficiency and higher  $\text{CO}_2$  emissions of these systems compared with grazing-only systems. The selection of the optimal or best production strategy is dependent on the relative value of each of these factors as determined by the whole of society.

The farming systems research at Karkedamm gives insight into the performance and N status of grassland systems across a range of N intensities. Additional studies have addressed (i) maize grown for silage (Jovanovic et al., 2000; Volkers et al., 2002) and (ii) a forage crop rotation including white clover–grass along with maize and cereals grown for silage (Bobe et al., 2004; Wachendorf et al., 2004b). This research provides a database for developing and calibrating models to extrapolate this information to other environments.

#### Improving Nutrient Utilization on Grassland Dairy Farms in the Netherlands

Dutch government policies implemented since 1985 (Henkens and van Keulen, 2001) have explicit objectives to restrict nutrient use in dairy farming. To comply with government policies while minimizing costs, dairy farmers need timely, relevant, and accurate information. For that purpose, the De Marke experimental dairy farm was established (Aarts et al., 1992). The strict environmental goals of the De Marke farm were based on long-term national environmental objectives that farms were to meet by 2020.

Bridging the gap in environmental performance between an experimental farm such as De Marke and commercial dairy farms requires coaching and the trans-

fer of knowledge. Therefore, a follow-up project called "Cows and Opportunities" was implemented to transfer and test the practical value of this management on 17 commercial pilot farms (Oenema et al., 2001).

### The De Marke Experimental Farm

The De Marke farm was located near Hengelo in the province of Gelderland on a sandy soil highly susceptible to nitrate leaching (Aarts et al., 2000a). A production system was developed to maintain a milk production of 12 000 kg ha<sup>-1</sup>, the average of dairy farms in this region during the mid-1980s. The farm was designed to minimize external inputs of feed and fertilizer and thus maximize the use of homegrown feeds and manure. The goal was a high milk production per cow (lower animal maintenance requirements to meet the farm milk quota) and a minimum number of calves and replacement heifers (Table 1) to reduce feed requirements per unit of milk produced. The proportion of grassland in the total crop area of De Marke was less than that found on most commercial farms with more forage maize produced. The low N (protein) concentration in maize was used to balance the high concentration in grass silage to formulate rations that efficiently met animal requirements and thus reduced N excretion. Moreover, water and fertilizer requirements per unit of grass DM produced were higher and energy yields lower than those of maize. The grassland area was greater than that of other forage crops though, to allow grazing and to better utilize manure nutrients.

Grazing was closely managed because of the increased risk of N leaching under pastures on this sandy soil. Grazing of lactating cows was restricted to 8 h d<sup>-1</sup> (changed to 4 h d<sup>-1</sup> in 2000) and in autumn cows were housed 1 mo earlier than that common in commercial practice. The grazing strategy used rotational stocking where paddocks were grazed for 4 d. These management practices reduced the number of urine and dung patches in the pasture, the associated nitrate leaching from these "hot spots," and yield losses from trampling. In addition, more manure nutrients were collected in the barn for crop use, thus reducing the external fertilizer requirement.

The farm area consisted of 11 ha of permanent grassland and 44 ha of rotated grass and maize. Rotated fields were in grass for 3 yr followed by 3 to 5 yr of maize. Rotation was used to stimulate maize growth and to avoid a high build-up of organic matter with the associated risk of nitrate leaching following decomposition when the sod was plowed. In the first year after grass, maize was not fertilized; decomposition of the plowed grass sod provided sufficient mineral N. Each year, when the maize was at a height of about 60 cm, Italian ryegrass [*Lolium multiflorum* (Lam.)] was sown between the maize rows. As the maize crop matured and after harvest, this cover crop took up excess fertilizer and mineralized N at a rate of about 110 kg ha<sup>-1</sup>. In early spring, the ryegrass sward was plowed providing nutrients for the subsequent maize crop. On commercial farms, both maize and grass are normally grown without rotation, and use of a cover crop after maize is rare.

**Table 1. Characteristics of the De Marke experimental farm, compared with those of the average Dutch farm in the middle of the 1990s with similar milk production and soil type (Aarts et al., 2000a).**

Characteristics	De Marke	Commercial farm
Cows ha <sup>-1</sup>	1.4	1.6
Young stock ha <sup>-1</sup>	1.0	1.5
Grazing season	1 May–1 October	1 May–1 November
Daily grazing (h, average season)	8†	14
Grass area: maize area‡	55: 45	75: 25
(Re)sowing of grassland	Spring–early summer	autumn
Fertilization period for grassland	1 March–15 August	1 February–1 September
Crop rotation	yes	no
Cover crop after maize	yes	no

† From 2000 onward, grazing time was restricted to 4 h d<sup>-1</sup>.

‡ From 2000 onward, the last year of maize in each rotation was replaced by triticale (silage) with a grass/clover mixture sown between the triticale rows.

Part of the maize was harvested as ground ear silage, which was used as a concentrate substitute in cattle rations. A special machine was used to harvest the ear silage and maize stover simultaneously, with each handled and stored separately. Energy, protein, and K concentrations of maize stover were low and fiber concentration was high, making it a suitable feed for dry cows and older heifers when blended with fall-harvested grass silage. Maize forage yields were higher than those of grass, and on the drought-sensitive soils of De Marke, less irrigation was required. Hence, a larger proportion of maize in the rotation reduced both the groundwater required for irrigation and the need for purchased feed. Furthermore, a high proportion of maize (with high energy and low protein concentrations) in the feed ration reduced the nutrient contents in the manure produced. Since 2000, the last year of maize in a rotation was replaced with triticale (*× Triticosecale* Wittmack) sown in autumn, interseeded with a grass–clover mixture between the rows. This management practice was introduced to better accommodate the timing of operations and to further reduce nitrate leaching loss.

Nitrogen fertilization levels at De Marke, including N from slurry, clover, and the residue of plowed-under Italian ryegrass and grass sod, were about 40% lower than those on commercial farms. About 75% of the slurry produced (containing about 3.5 kg N m<sup>-3</sup> and 0.46 kg P m<sup>-3</sup>) was applied to grassland by shallow injection in two to three applications with a total of 53 m<sup>3</sup> ha<sup>-1</sup> on permanent grassland and 74 m<sup>3</sup> ha<sup>-1</sup> on rotated grass. Maize was fertilized with an average of 23 m<sup>3</sup> ha<sup>-1</sup> of manure by a single deep injection before sowing. Additional inorganic N fertilizers were applied on grassland at a rate of 107 kg N ha<sup>-1</sup>. Slurry and other fertilizers were applied between 1 March and 15 August to reduce the risk of nitrate leaching in the autumn and winter when low temperature, solar radiation, and crop growth result in surplus precipitation.

All N and P inputs, outputs and flows through the major farm components were measured or estimated at De Marke during each year of operation. From these data, a complete farm balance and the transfer of these nutrients through the major farm components was es-

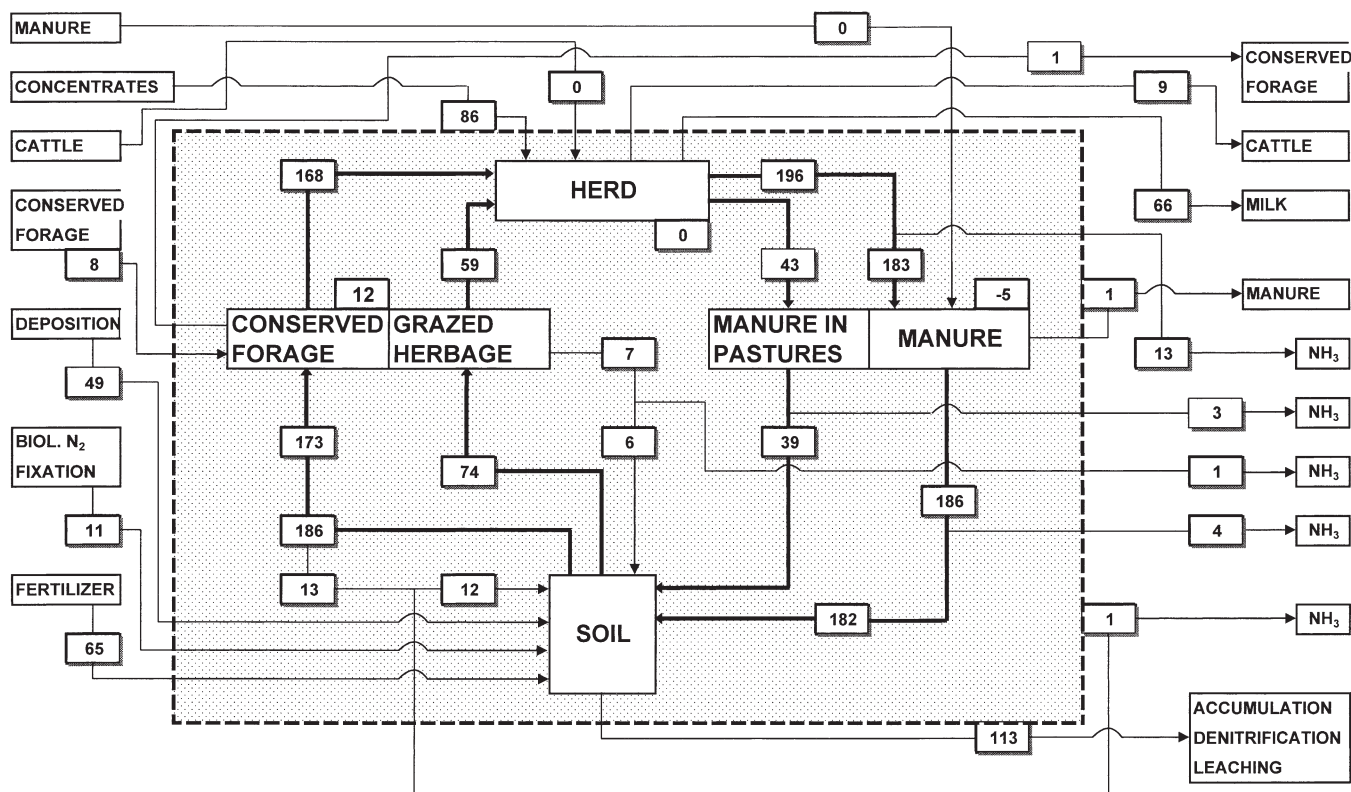


Fig. 6. Average annual nitrogen inputs, outputs, and flows ( $\text{kg N ha}^{-1}$ ) through the major components of the De Marke experimental farm in the Netherlands for 1993 to 2002. Adapted from Aarts et al. (2000a) and Hilhorst et al. (2001).

established (Fig. 6 and 7). The nutrient balance around each of the major farm components determines the efficiency in nutrient utilization and thus identifies the most inefficient processes on the farm. Strategic management

changes can then be made to improve the nutrient use efficiency within the farm and further reduce losses to the environment.

Total surplus N for the farm included an accumulation

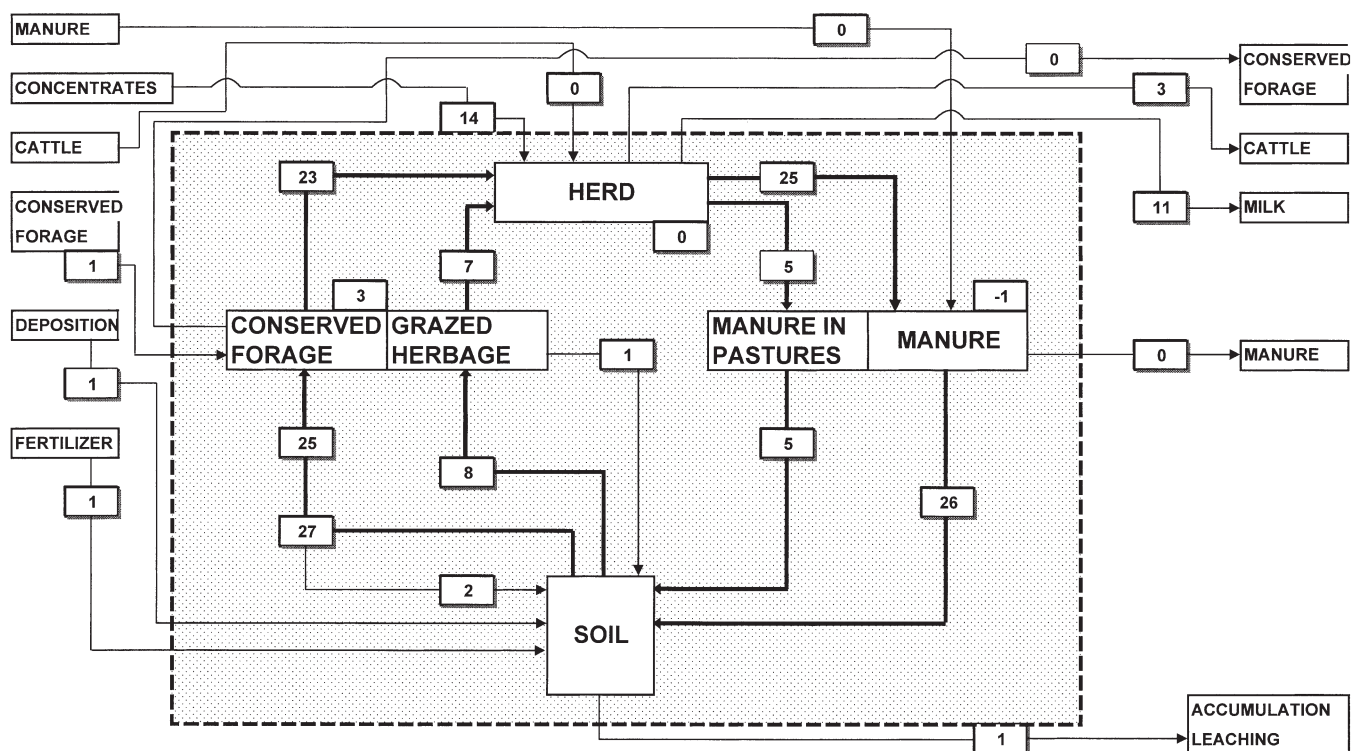


Fig. 7. Average annual phosphorus inputs, outputs, and flows ( $\text{kg P ha}^{-1}$ ) through the major components of the De Marke experimental farm in the Netherlands for 1993 to 2002. Adapted from Aarts et al. (2000b).

**Table 2.** Nutrient balances (N and P) of the De Marke experimental farm averaged over the period 1993–2002 and for the year 2002, compared with the balances of the average Dutch farm in the middle of the 1990s (source: Hilhorst et al., 2001).

compared with the balances of the average Dutch farm in the middle of the 1990s (Source: Timmerik et al., 2001).						
Item	Nitrogen			Phosphorus		
	De Marke		Average	De Marke		Average
	1993–2002	2002	Farm	1993–2002	2002	Farm
	kg N ha <sup>-1</sup>			kg P ha <sup>-1</sup>		
Input						
Concentrates	86	87	125	12	13	21
Roughage	8	0	20	1.3	0	0.9
Chemical fertilizer	64	35	242	0.4	0	18
Organic manure	0	0	50	0	0	13
Biological N fixation	11	27	0	0	0	0
Animals	0	0	0	0	0	0
Deposition	49	49	49	0.9	0.9	0.9
Miscellaneous	5	5	0	0	0.4	0
Total	223	203	486	15	14	54
Output						
Milk	66	64	64	10	10	10
Animals	9	8	14	3	2	4
Roughage	1	0	0	0	0	0
Organic manure	1	0	0	0	0	0
Total	77	72	78	13	12	15
Changes in stocks	2	14	0	0.4	0.4	0
Surplus	144	117	408	1.3	1.3	39

in soil organic matter and losses through NH<sub>3</sub> volatilization, denitrification, leaching, and runoff (Table 2). Losses in runoff were negligible because of little surface water movement. Average annual surplus from 1993 to 2002 was 144 kg N ha<sup>-1</sup> and 1.3 kg P ha<sup>-1</sup>. The design of the farming system was modified in 2000 (shorter grazing periods and reduced fertilization), which reduced the N surplus to 117 kg N ha<sup>-1</sup> by 2002. A comparison of the nutrient balance of De Marke to that of an average commercial farm (on sandy soil in the mid 1990s with a similar milk quota) shows that at De Marke, less fertilizer and feed were purchased (Table 2). In other words, maintaining high nutrient use efficiencies in animal nutrition and crop cultivation allowed similar milk production with a lower level of nutrient input. The economics of both systems are discussed in De Haan (2001).

### Commercial Pilot Farms

Innovative and possibly risky farm designs can be tested, adjusted, and improved on experimental farms. In practice, dairy farmers are often reluctant to adjust management because of a lack of information and a lack of confidence in the proposed innovations. Intensive coaching and regular interaction among researchers, advisory service personnel, and farm owners can build confidence and accelerate adoption of efficient nutrient management systems on pilot farms. If these pilot farmers are respected by their peers, knowledge can be transferred quickly and reliably. Thus, pilot farms should represent the full range of dairy farms in the region, and the farmers should be among the best managers (Reijneveld et al., 2000). In the project "Cows and Opportunities," 17 commercial farms were selected in the Netherlands (primarily on dry sandy soils) as pilot farms to adopt the nutrient management practices developed at the De Marke experimental farm (Oenema et al., 2001).

After collecting detailed information, each participating farm was analyzed to identify strengths, weaknesses,

and opportunities for improvement. This analysis also identified the gap between target and actual nutrient surpluses (Oenema et al., 2000). Target surpluses were farm specific, depending on soil type, hydrology, cropping pattern, and level of milk production. Subsequently, a farm development plan was formulated for each participant. Farmers had a strong influence on the plan, but expected (model-calculated) nutrient surpluses could not exceed permitted levels. Next, farm action plans were developed and implemented in 1999. Each subsequent year, the plans were modified on the basis of the performance of the preceding year.

The pilot farms were characterized by higher milk quota (48%), more total land area (36%), more land planted in maize (15%), and greater milk production per cow (15%) than the average Dutch farm. Before the project started in 1998, the pilot farms had an average farm area of 41 ha, a milk quota of 14 300 kg ha<sup>-1</sup>, 69 milking cows with 55 head of young stock, and a milk production of 8000 kg cow<sup>-1</sup>. In the course of the project, these farms increased in size, intensity, and milk production per cow. Thus, differences between the pilot and traditional farms increased further. Pilot farmers exploited the available knowledge provided by research and advisory service staff and other pilot farmers to improve their personal skills.

Nitrogen fertilizer rates decreased substantially during the project, while milk production increased (Table 3). This reduction was achieved through reduced grazing time (more manure collected for spreading) and less chemical fertilizer use. A fertilizer reduction of more than 100 kg N ha<sup>-1</sup> on grassland contributed most to this overall reduction. A large range in fertilization rates was found on all farms. On grassland, manure application rates ranged from 106 to 366 kg N ha<sup>-1</sup> with inorganic fertilizer rates from 0 to 297 kg N ha<sup>-1</sup>.

Grassland yields from 1999 to 2002 averaged 11.5 Mg DM ha<sup>-1</sup> and ranged from 7 to 16 Mg DM ha<sup>-1</sup>. This variation was associated with differences in soil type, fertilization rates, crop management, and weather con-

**Table 3. Nitrogen fertilization on the commercial pilot farms in the “Cows and Opportunities” project for 1998 and 2002, including manure organic N.**

N input	1998		2002	
	average	range	average	range
kg N ha <sup>-1</sup>				
<b>Farm level</b>				
Applied organic manure	204	102–322	209	95–311
Pasture manure	73	10–142	40	0–73
Chemical fertilizer	173	0–297	86	0–137
<b>Grassland</b>				
Applied organic manure	226	106–366	236	104–351
Pasture manure	98	15–182	57	0–123
Chemical fertilizer	222	0–297	108	0–185
<b>Other crops</b>				
Applied organic manure	172	0–310	141	0–262
Chemical fertilizer	50	0–180	46	0–144

ditions. About two-thirds of the total production was harvested as silage with the remainder grazed. Some farmers harvested all grass as silage with no grazing, whereas others used extended grazing periods. Silage maize yield averaged 14 Mg DM ha<sup>-1</sup> and ranged from 8 to 20 Mg DM ha<sup>-1</sup>.

In the course of the project, average farm surpluses of N and P were reduced by 32 and 50%, respectively, mainly through reduced fertilizer use (Table 4). Export of animal manure decreased from 23 to 9 kg N ha<sup>-1</sup>, indicating improved utilization of farm-produced manure. Reductions in the purchase of feeds and fertilizers improved the economic performance of the pilot farms. On the most intensive farms, net reductions in production costs of \$0.006 kg milk<sup>-1</sup> (\$3630 farm<sup>-1</sup>) were attained after accounting for any extra management costs. Additional expenditures for attaining these reduced surpluses were negligible, but farmers spent more time planning fertilization and feeding strategies, and recording and analyzing farm data.

### Lessons Learned from the Dutch Work

Combining the experimental prototype farm and the pilot farm evaluation produces important benefits (Verijken, 1992). This combined approach focuses on the farm level, the level at which management decisions are

**Table 4. Average nutrient balances for the pilot farms in the “Cows and Opportunities” project for 1998 and 2002.**

Item	N		P	
	1998	2002	1998	2002
— kg N ha <sup>-1</sup> —				
<b>Input</b>				
Animals	1	0	0.4	0
Organic manure	9	9	2	2
Chemical fertilizer	178	84	13	4
Clover	4	8	0	0
Deposition	43	44	0.9	0.9
Concentrates	123	109	20	18
Forage	26	26	4	4
Total	383	280	39	28
<b>Output</b>				
Milk	78	77	13	13
Animals	14	14	4	4
Organic manure	23	9	4	2
Forage	2	4	0.4	0.9
Total	117	105	21	19
Surplus	266	176	18	9

made. Through this approach, processes are integrated and knowledge transfer is facilitated. By bridging theory and experiments, the risk of giving too much attention to those elements less relevant in practice is avoided or, vice versa, that insufficient attention is given to issues that prove important later.

Working with experimental and pilot farms also has disadvantages. One experimental system must be selected, rather subjectively, from a number of options. A reliable comparison with other systems is impossible because only one system can be implemented and even that system is continuously evolving. Therefore, results cannot be statistically tested. These disadvantages can be partly overcome by (i) using an appropriate monitoring program, (ii) conducting additional interdisciplinary research, aimed at determining causal relations that explain the behavior of the system and used to investigate the consequences of alternatives (e.g., farming systems work such as at the Karkendamm farm), and (iii) combining the experimental research of the prototype farm with modeling to explore alternative possibilities (Van de Ven and van Keulen, 1996; Hack-ten Broeke, 2000).

### Simulation Analysis of Grassland Farming Systems

Computer simulation provides a tool for rapid and inexpensive evaluation of the long-term performance of farming systems. With computer simulation, many variants of the production system can be easily evaluated, including different climatic regions, soil types, and farm management scenarios. A limitation of this approach is developing confidence in the simulation results. Only through extensive verification and evaluation of the model, can the model user and others using the model-generated information become confident in the results.

A combination of experimental and modeling evaluations provides the most comprehensive approach. Measured data and information from actual production systems provide a basis for calibrating and evaluating a model. The simulation model can then be used to extrapolate site-specific information to other situations.

### Whole-Farm Models

A number of models have been developed and applied to the evaluation of livestock production in grassland farming systems. Although these models function at the farm level, most do not include all major farm components, or these components are not modeled with enough detail to provide a robust research and teaching tool. Most models emphasize either grazing systems or conserved forage systems with few including both on the same farm.

Most farm models may be classified as either linear programming or simulation models. Linear programming is often used for farm economic evaluation and optimization. This approach has been used to evaluate crop and sheep production in western Australia (Pannell, 1995) and dairy production in New Zealand (McCall and Clark, 1999), the USA (Schmit and Knoblauch,

1995), and the Netherlands (Berentsen and Giesen, 1995; Van de Ven and van Keulen, 1996; Pacini, 2003). Although these models are primarily used for economic evaluation, environmental issues have also been addressed by tracking N, P, and K balances at the soil, animal, and farm levels (Berentsen and Giesen, 1995). A limitation in linear programming is that the dynamics of the biological and physical processes within the farm cannot be adequately described to incorporate their effects and interactions on farm performance.

Comprehensive dynamic simulation models, that follow the major farm processes through time, typically provide more robust and adaptable research tools for evaluating production systems. Process-level models have been used to evaluate the bio-economic efficiency of beef production in Canada (Pang et al., 1999), rangeland beef production in the western USA (Carlson and Thurow, 1996), pasture beef production in the eastern USA (Parsch and Loewer, 1987), and pasture dairy production in France (Cros et al., 2003). A simulation-based decision support system has also been developed to provide crop and livestock management support at the whole-farm or ranch level, with emphasis on water, nutrient, and pesticide management (Ascough et al., 2001).

Simulation models of forage growth, harvest, storage, and use have also been used to evaluate confinement dairy systems. Models developed in Europe used grass-based forage systems alone (McGechan, 1990) or in combination with wheat (Doyle et al., 1990). The Dairy Forage System Model (DAFOSYM), developed in the USA, emphasized the use of alfalfa and maize crops (Rotz et al., 1989). These comprehensive models provided tools for evaluating and comparing the performance and economics of various technologies and management strategies for dairy farms (McGechan and Cooper, 1995; Rotz et al., 1999b; Rotz et al., 2002). In addition to this public domain software, at least two simulation models are commercially marketed for use in dairy farm planning and evaluation (Dexcel, 2004; Larcombe and McLean, 2004).

Emphasis in the development and application of farm models recently shifted from performance and economics toward nutrient cycling and environmental impact. Several simulation studies in the Netherlands evaluated N loss from dairy farms using information obtained from De Marke (Hack-ten Broeke and van der Putten, 1997; Hack-ten Broeke and de Groot, 1998; Hack-ten Broeke et al., 1999; Kuipers et al., 1999; ten Berge et al., 2000). Topp and McGechan (2003) extended their Scottish farm model to include N dynamics and then evaluated the impacts of varying fertilizer application rates and stocking densities on nitrate leaching, denitrification losses, and net farm income. Recent work has expanded DAFOSYM to a whole-farm model with similar emphasis on performance, economic, and environmental issues. This new version, called the Integrated Farm System Model (IFSM), provides a comprehensive and robust farm model for evaluating dairy and beef production in grassland systems (Rotz et al., 2005).

## The Integrated Farm System Model

The Integrated Farm System Model assimilates the many biological and physical processes on dairy and beef farms (Rotz et al., 1999b; Rotz et al., 2005). Crop production, feed use, and the return of manure nutrients to the land are simulated over many years of weather. Growth and development of grass, alfalfa, maize, soybean, and small grain crops are determined on a daily time step as a function of soil and weather conditions. Tillage, planting, harvest, and storage operations are simulated to predict resource use, timeliness of operations, crop losses, and nutritive changes in feeds. Feed allocation and animal response are related to the nutritive value of available feeds and the nutrient requirements of the animal groups making up the herd (Rotz et al., 1999a).

Nutrient flows through the farm are modeled to predict potential nutrient accumulation in the soil and loss to the environment. The quantity and nutrient content of the manure produced is a function of the quantity and nutrient content of the feeds consumed. Nitrogen volatilization occurs in the barn, during storage, following field application, and following grazing. Denitrification and leaching losses from the soil are related to the rate of moisture movement through the soil profile as influenced by soil properties, rainfall, and the amount and timing of manure and fertilizer applications. Whole-farm balances of N, P, and K consider the import of nutrients in feed and fertilizer and the export in milk, animals, and excess feed.

Simulated performance is used to predict production costs, income, and farm net return or profit for each year of weather. A whole-farm budget is used where investments in equipment and structures are depreciated over their economic life, and the resulting annual costs are added to other annual expenditures and incomes determined for each year. By simulating various production alternatives, the effects of system changes are compared with respect to resource use, production efficiency, environmental impact, and net return. The distribution of annual values is used to assess the risk involved in alternative technologies or strategies as influenced by weather. Further detail on the algorithms and assumptions used in the model can be found in the reference manual (Rotz and Coiner, 2003).

Simulated production systems have been evaluated against information collected on commercial farms in the USA, the Karkendamm experimental farm in Germany, and the De Marke farm in the Netherlands. The model has then been used to evaluate nutrient conservation technologies and management strategies tested on these farms under other farm, climatic, and soil conditions.

## Evaluation of Grassland Farming Systems in the USA

DAFOSYM, and more recently IFSM, has been used to evaluate and compare a number of grassland management options in the USA. For example, simulations of a grass farm in Pennsylvania demonstrated that combining grazing with supplemental grain feeding reduced

N loss and the accumulation of soil P compared with traditional confinement systems or grazing systems with little or no grain feeding (Soder and Rotz, 2001). The decreased N loss appears to contradict the research in Europe where grazing time was reduced to reduce N loss. However, the sandy soils in northern Europe and the technology used to reduce N loss during manure handling lead to proportionally more N loss from grazing animals under their conditions. On the simulated Pennsylvania farm, profit increased as grain supplementation increased to an annual net return \$330 cow<sup>-1</sup> greater than that of the confinement system. In another farm study in Pennsylvania, use of grazing as a supplement to the feeding of total mixed rations using farm-grown maize and alfalfa silages resulted in a small decrease in farm profit with little environmental impact compared with full confinement feeding (Soder and Rotz, 2003).

In a study to examine options for reducing P loading on New York dairy farms, greater use of grass and more intensive grazing of pastures increased farm net return, while maintaining or reducing soil P accumulation (Rotz et al., 2002). Implementation of a rotational grazing strategy on a 100-cow farm increased annual net return about \$100 cow<sup>-1</sup>. Conversion of all cropland to grass, along with more intensive use of grazing, further increased this net return by \$24 cow<sup>-1</sup>. On an 800-cow farm, higher N fertilizer application and more intensive use of grassland increased the annual net return by \$80 cow<sup>-1</sup>, with small reductions in N leaching loss and soil P accumulation.

#### Application of IFSM to the Karkendamm Farm

We used IFSM to extrapolate the N cycling data measured on the Karkendamm farm to whole farm systems. The first step in this process was to compare N loss values predicted by the model to those measured. The 64 crop-management scenarios used at Karkendamm were simulated over the weather of 1997 to 2000. Forty scenarios for grassland production included four levels of N fertilization and five defoliation methods, all with and without the application of dairy manure slurry. The remaining 24 scenarios for maize included four mineral N fertilization rates and three manure application rates, all with and without the use of a grass cover crop to take up and retain soil N following maize harvest.

The model predicted leaching losses over this wide range of management scenarios with reasonable accuracy (Fig. 8). Prediction error that occurred involved both measurement error and model inadequacies. Important conclusions of this comparison were that the model was able to predict similar annual losses as those measured over a wide range in crop and fertilization conditions and that model predicted trends were more consistent across management scenarios than those determined by measurement. Because of the sandy soils, high N fertilization rates, and high stocking rates used in this experiment, much greater N leaching losses were often measured on the grassland than those measured under maize.

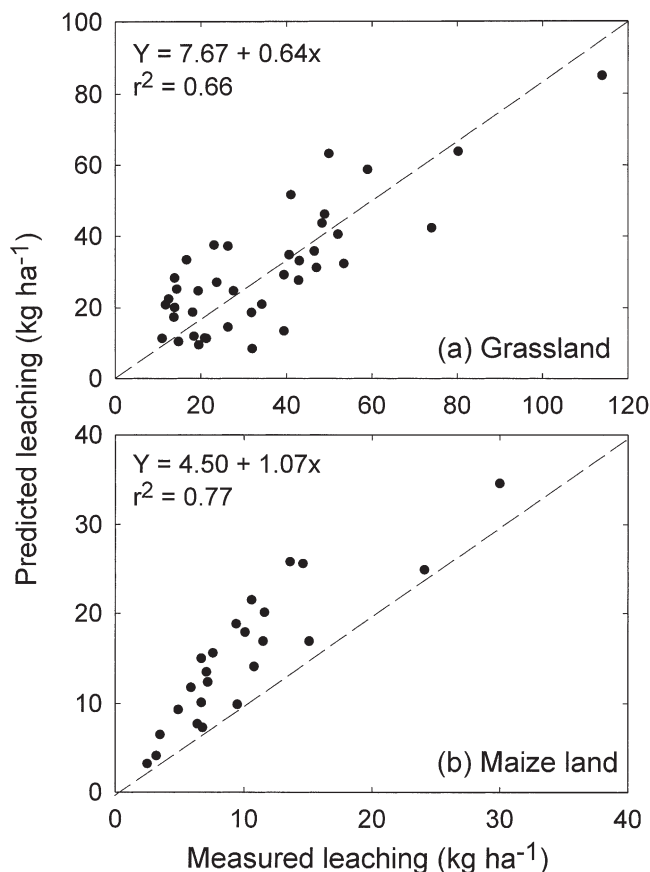


Fig. 8. A comparison of simulated and actual nitrogen leaching losses from grass (a) and maize (b) plots at the Karkendamm Research farm over a wide range in fertilizer application rates and defoliation methods.

With the farm model, experimental results of the Karkendamm study can be applied to whole farm systems where both grass and maize are produced. To illustrate this approach, a representative (hypothetical) dairy farm with the characteristics of farms in this region was simulated for the years of 1980 to 2002 using weather data for Kiel, Germany. The farm included 55 cows and 48 replacement heifers on 34 ha of loamy sand soil. Four management scenarios were simulated on the farm: (i) all land in grass with the grass harvested and fed as silage; (ii) all grass with one-half grazed and the remainder harvested as silage; (iii) one-half grass and one-half maize, all harvested as silage; and (iv) one-half grass with grazing and one-half maize for silage. When maize silage (with its greater energy content) was fed on the farm, milk production was increased 6%.

Rotational grazing of the dairy herd had little effect on N import and export from the farm, but overall N losses were greater (Table 5). Annual N volatilization loss decreased about 8% and N leaching loss increased 11 to 40% with this increase directly related to the time spent grazing. Although grazing may reduce the homogeneity of soil P concentrations, it had little effect on the overall farm balance of P. Grazing reduced production costs and increased annual net return to farm management by \$127 cow<sup>-1</sup> when all older animals were grazed

**Table 5. Annual feed production, feed use, nutrient balances, production costs, and net return of a simulated dairy farm<sup>†</sup> in northern Germany with and without the use of grazing and maize silage.**

Production or cost parameter	Grass only <sup>‡</sup>		Grass and maize silage <sup>§</sup>	
	No grazing <sup>  </sup>	Grazed <sup>#</sup>	No grazing <sup>  </sup>	Grazed <sup>#</sup>
	Mg DM			
Grass silage production	278	144	149	82
Maize silage production	0	0	157	158
Grazed forage consumed	0	134	0	78
Forage purchased	81	97	50	51
Supplemental feed purchased	100	85	124	109
	kg cow <sup>-1</sup>			
Milk production	8000	8000	8500	8500
	kg ha <sup>-1</sup>			
Nitrogen cycled on farm <sup>††</sup>	476	494	397	415
Nitrogen imported	292	293	262	259
Nitrogen exported	83	83	87	87
Nitrogen lost by volatilization	80	73	64	60
Nitrogen lost by leaching	45	63	45	50
Nitrogen lost by denitrification	22	27	23	26
Soil phosphorus accumulation	9	9	12	10
Soil potassium accumulation	116	112	100	89
	\$ cow <sup>-1</sup>			
Feed cost	1279	1196	1333	1293
Manure handling cost	286	242	289	260
All other costs <sup>‡‡</sup>	1466	1466	1479	1478
Total production cost	3031	2904	3101	3031
Milk, feed and animal sale income	3731	3731	3949	3949
Net return to management	700	827	848	918
Standard deviation in net returns	60	59	70	71

<sup>†</sup> 55 cows and 48 replacement heifers on 34 ha of loamy sand soil simulated over years 1980 to 2002 using weather for Kiel, Germany.

<sup>‡</sup> 34 ha of grass with 100% of the total forage requirement for the herd obtained from grass silage and pasture.

<sup>§</sup> 19 ha of grass and 15 ha of maize with 50% of the total forage requirement for the herd obtained from maize silage and the remainder from grass.

<sup>||</sup> Entire grass crop is harvested, conserved and fed as silage.

<sup>#</sup> About 50% of the annual grass forage consumption is fed through grazing.

<sup>††</sup> Average N cycled through the farm each year from manure, fertilizer, legume fixation, and deposition.

<sup>‡‡</sup> Includes annual costs of milking and housing facilities, livestock expenses, milk transport, milking labor, and property tax.

(Column 2, Table 5) and \$70 cow<sup>-1</sup> when lactating cows were grazed half days (Column 4, Table 5).

Use of maize on the farm resulted in greater feed production, more efficient use of supplemental feed, and a reduction in the use of N fertilizer (Table 5). Less fertilizer N was used with maize because high mineral N application rates are typically used for grassland production on the sandy soils in this region. This resulted in a 20% reduction in volatile N loss from the farm, with little effect on N leaching and denitrification losses. Soil P accumulation increased slightly because of greater use of purchased supplemental feeds to meet the herd's protein requirements. Production costs were higher with maize production, but this was more than offset by the increase in farm income through greater milk sales. Net return to farm management was \$148 cow<sup>-1</sup> higher with maize production on the nongrazing farms and \$91 cow<sup>-1</sup> higher on grazing farms. Maize production increased year-to-year variability in farm net return, due primarily to the greater variability in maize yield relative to grass.

### Application of IFSM to the De Marke Farm

We also applied IFSM to evaluate the feasibility of the nutrient conservation technologies and management strategies used at De Marke. First, we compared the environmental results predicted by the model to actual De Marke farm data, and then used the model to assess these technologies on other farms. To evaluate the model, parameters were set to represent the crop, ma-

chinery, harvest, animal, and manure handling characteristics of De Marke. First, 4 yr were simulated from 1996 to 1999 when production practices were relatively constant with only grass and maize produced on the farm (Hilhorst et al., 2001). The following 2 yr were then simulated where the original 24.5 ha of maize were reduced to 18 ha of maize plus 6.5 ha of triticale. Grazing of the milking herd was reduced to 4 h d<sup>-1</sup> with a 50% reduction in grazing area. The legume content in the grass sward was also increased slightly to reflect an observed higher proportion of clover in the stand.

Average annual simulated feed production and use, and N and P flows compared very closely to observed values. Nitrogen imports from fertilizer, feed, deposition, and fixation were all closely represented by the model with total import within 2% of the actual reported values (Fig. 9). Losses through volatilization, leaching, and denitrification, and the N export in milk and animals sold were all within 10% of actual values. Over all 6 yr, the accumulated or unaccounted soil N simulated by the model was only 2% (0.8 kg N ha<sup>-1</sup>) less than that determined for the actual farm. This evaluation supported that the model was able to reproduce the N and P flows of this well-managed experimental farm.

To determine the long-term benefits of the management strategies and technologies used at De Marke, three production systems were compared for a representative dairy farm of this region (Reijneveld et al., 2000; Aarts et al., 1999). The farm included 55 cows and 48 replacement heifers on 34 ha (26 ha of grass and 8 ha

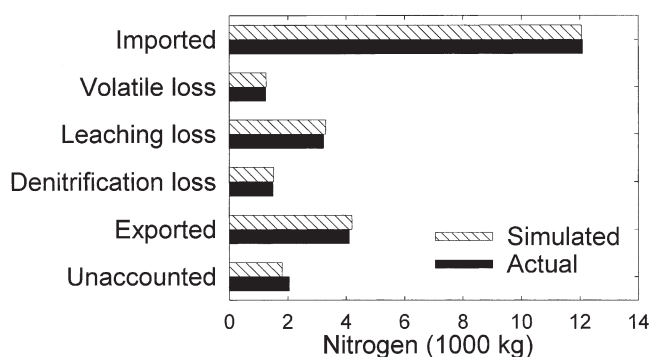


Fig. 9. A comparison of simulated and actual nitrogen balances for the De Marke dairy farm for weather years 1996 to 2001.

of maize) of loamy sand soil simulated over 1977 to 2001 weather for Wageningen, the Netherlands. The three production systems represented previous technology, current technology, and the De Marke technology for nutrient conservation (Aarts et al., 1992). Previous technology portrayed a farm with little interest in nutrient conservation. An open manure storage was used with broadcast application of manure, and 275 kg N ha<sup>-1</sup> of mineral fertilizer was applied to grassland. Animals were grazed about 16 h d<sup>-1</sup> and maintained an annual milk production of 8000 kg cow<sup>-1</sup>. The current technology reflected recent changes on Dutch dairy farms to improve N use efficiency. These included an enclosed manure storage, manure application by injection, and 175 kg N ha<sup>-1</sup> of mineral fertilizer applied to grassland. Animals were grazed 8 h d<sup>-1</sup>, maintaining a milk production of 8500 kg cow<sup>-1</sup>. The De Marke technology included low emission barn floors with feces and urine separation, an enclosed manure storage, manure application by injection, 120 kg N ha<sup>-1</sup> of mineral fertilizer applied to grassland, and a grass cover crop following maize (Aarts et al., 2000a, 2000b). Farm area was increased by 9 ha of maize, which was harvested for both ear silage and stover silage.

The production system under previous technology reflected inefficient use and cycling of N. Excessive amounts of N, primarily in the form of mineral fertilizer, were imported and cycled through the farm, causing high losses to the environment (Table 6). A comparison of current and previous technologies indicates both positive and neutral environmental impacts for recent changes in the Dutch dairy industry. A large reduction in N volatilization loss was obtained using the enclosed manure storage and manure injection; however, there was little effect on nitrate leaching and soil denitrification losses and the accumulation in soil P. The reduction in volatile N loss led to high levels of soil N, even with a reduction in the use of N fertilizer.

Only by fully implementing the practices of De Marke, were substantial improvements in N-use efficiency achieved with a large reduction in the import of N in fertilizer and feed. Nitrogen volatilization losses were greatly reduced along with 50% reductions in leaching and denitrification losses (Table 6). Long-term soil P accumulation was also eliminated. The challenge of this approach is maintaining farm profitability. The simula-

Table 6. Effect of using technologies for nitrogen conservation on annual feed production, feed use, nutrient balances, production costs, and net return of a simulated dairy farm in the Netherlands†.

Production or cost parameter	Previous technology‡	Current technology§	De Marke technology¶
<b>Mg DM</b>			
Grass silage production	129	173	187
Maize silage production	81	81	99
Maize ear silage production	0	0	40
Grazed forage consumed	127	65	63
Forage purchased	10	31	10
Supplemental feed purchased	114	124	95
<b>kg cow<sup>-1</sup></b>			
Milk production	8000	8500	9000
<b>kg ha<sup>-1</sup></b>			
Nitrogen cycled on farm#	598	519	378
Nitrogen imported	382	318	214
Nitrogen exported	89	89	76
Nitrogen lost by volatilization	116	53	26
Nitrogen lost by leaching	84	82	46
Nitrogen lost by denitrification	39	37	23
Phosphorus accumulation	5	6	0
Potassium accumulation	42	71	43
<b>\$ cow<sup>-1</sup></b>			
Feed cost	1228	1327	1671
Manure handling cost	209	275	308
All other costs††	1468	1479	1506
Total production cost	2905	3050	3485
Milk, feed and animal sale income	3789	3954	4180
Net return to management	884	873	695
Standard deviation in net returns	60	64	87

† 55 cows and 48 replacement heifers on 34 ha (26 ha of grass and 8 ha of maize or 17 ha of maize with De Marke technology) of loamy sand soil simulated over years 1977 to 2001 using weather for Wageningen, the Netherlands.

‡ Includes standard barn floor, bottom loaded 6-mo manure storage, broadcast application, and full-day grazing.

§ Includes standard barn floor, enclosed 6-mo manure storage, injection application, and half-day grazing.

¶ Includes 9 ha more maize land harvested as high-moisture ear maize and stover, low fertilizer use, a grass cover crop following maize, low emission barn floor with feces and urine separation, an enclosed 6-mo manure storage, and manure application by injection.

# Average N cycled through the farm each year from manure, fertilizer, legume fixation, and deposition.

†† Includes annual costs of milking and housing facilities, livestock expenses, milk transport, milking labor, and property tax.

tion results imply that implementation of all of these practices for improving nutrient conservation increased annual production costs by \$580 cow<sup>-1</sup>. Increased income through greater milk production offset a portion of the increased costs, but a \$190 cow<sup>-1</sup> loss in the annual net return to farm management still occurred (Table 6).

## Conclusions

Nutrient cycling in grassland agriculture is fundamentally different from that in row-crop agriculture. In pastures, livestock gather forage from large areas and excrete unused nutrients in concentrated patches, creating greater heterogeneity in nutrient availability. Although these concentrated nutrient patches increase the risk of nutrient loss, well-managed perennial grasslands also provide inherent capacities to reduce adverse environmental effects whether harvested or grazed. In many areas of North America and Europe, grasslands are managed along with annual crops such as maize forming very complex livestock farming systems. A whole-farm approach is needed to develop more efficient nutrient management strategies that reduce nutrient losses to

the environment while maintaining profitable production systems.

Experimental farm research in northern Europe has documented the major nutrient flows in grassland farming systems under specific soil and climate conditions. Farm management changes are being demonstrated that reduce nutrient inputs, improve nutrient cycling within the farm, and thus reduce losses to the environment. A pilot farm program is transferring techniques and strategies proven on the experimental farms to use on commercial farms.

Evaluation and adaptation of these technologies and strategies to farms with different soil and climate conditions requires expensive long-term research when experimental farms are used. Computer simulation supported by field and farm studies provides another relatively rapid and cost-effective approach. When properly evaluated on experimental farms such as those in northern Europe, a process-level farm simulation model provides a powerful tool for evaluating and refining production systems on farms in other locations. One such tool, the Integrated Farm System Model, is being used to develop, evaluate, and promote more sustainable grassland systems for commercial dairy and beef production in temperate regions of the world. Simulation of dairy farms on the sandy soils of northern Europe illustrate that maize silage can be used along with grasslands to increase farm profitability while maintaining or reducing nutrient loss to the environment. Technologies such as cover crops, low-emission barns, covered manure storages, and direct injection of manure into the soil can be used to greatly reduce N losses from farms, but their use creates a net cost to the producer.

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